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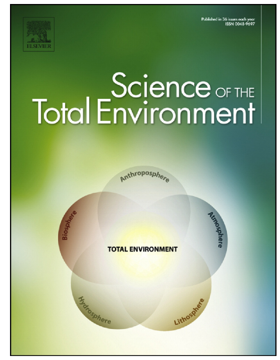

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The interplay between economics, legislative power and social influence examined through a social-ecological framework for marine ecosystems services

Simone Martino[1,2#], Paul Tett[1], Jasper Kenter[2]

[1] *Scottish Association for Marine Science, Laurence Mee Centre for People & the Sea, Oban, PA37 1QA*, UK

[2] *Environment Department, University of York, York, YO10 5DD, UK*

[#] Corresponding author: *Simone.Martino@sams.ac.uk; simone.martino@york.ac.uk;*

*c*o-authors: *Paul.Tett@sams.ac.uk; jasper.kenter@york.ac.uk*

Abstract

In the last 15 years, conservation has shifted increasingly towards perspectives based on the instrumental value of nature, where what counts is what provides benefits to humans. The ecosystem services framework embraces this vision of nature through monetary valuation of the environment to correct market failures and government distortions that hinder efficient allocation of public goods, including goods and services provided by biodiversity and ecosystems. The popularity of this approach is reflected in different countries legislation; for instance, US, EU and UK have introduced economic criteria for comparing costs and benefits of environmental policies in protecting ecosystem services.

From an operational perspective, the ecosystem services framework requires ecologists to estimate how the supply of services is affected by changes in the functionality and/or the extent of ecosystems; and economists to identify how changes in the supply affect the flow of direct and indirect benefits to people. However, this approach may be simplistic when faced with the complexity of social-ecological systems. We investigated this for three different marine services: assimilative capacity of waste, coastal defense and renewable energy. We find that economic valuation could provide efficient and fair allocations in the case of assimilative capacity, but leads to social clashes between outputs generated by cost benefit analysis and citizens' expectation in the case of coastal defense. In the case of renewable energy, controversies can be generated by regulatory mechanisms that are not necessarily aligned with the interests of industry or important social groups. We conclude that there is a need to integrate perspectives arising from utilitarian allocation of resources with those involving legislation and communal values in order to reconcile conflicting interests and better sustain marine social-ecological systems.

Keywords: marine ecosystem services, economic valuation, legislation, social-ecological systems, complexity

Highlights

1. A social-ecological framework analyses three marine ecosystem services (ES)
2. Assimilative capacity, coastal defence and renewable energy are object of study
3. The role of economics, regulation and civil society is debated for the three ES
4. Economics shows some axiomatic issues in guaranteeing the provision of these ES
5. Other institutions are required to enable social learning and conflicts resolution

1. Introduction

Ecosystem-based management principles (Garcia et al. 2003) have led to new forms of governance and management of natural resources. In terms of coastal and marine resources, there has been a proliferation of new paradigms, such as those for ecosystem-based fisheries management (Garcia et al. 2003; Staples et al. 2014), Integrated Coastal Zone Management, Marine Spatial Planning and the Ecosystem Services framework within public policy processes (Fisher et al. 2009). From an operational perspective, the ecosystem services framework requires ecologists to estimate how ecosystems supply services supported by complex ecological functions and processes; economists and other social scientists to identify how changes in ecosystem services provision affect the flow of direct and indirect benefits; and ecologists and economists to work together to define how services are provided under the complex interactions of social-ecological systems. For policymaking purposes, the ecosystem services framework can be seen as a tool for integration of environmental, social and economic knowledge (Danley and Widmark 2016). It can be used to conceptualise the social system as the human users of the ecological system (Binder et al. 2013) and contributes to the renaissance of the conservation paradigm "nature for people" (Mace 2014), where decisions are derived on the basis of instrumental support (generation of wellbeing) from nature to humans.

The broad diffusion of the ecosystem services framework was promoted by the surge of monetary valuations that have in turn stimulated implementation of finance mechanisms based on nature commodification (OECD 2013). This is reflective of a perceived increased influence of economic decisions in policy mechanisms (Pearce et al. 2006; Carey 2014): in those contexts where system uncertainty is limited, and decisions are top-down, monetization has become the common denominator to discriminate positive and negative impacts of decision-making on human wellbeing, and Cost Benefit Analysis (CBA) the tool of choice to classify good and bad projects, and policies through a technocratic approach aimed at maximising social welfare (Funtowicz and Ravetz 1993; 1994). These monetary approaches are part of a utilitarian, efficiency-based way of thinking grounded in the moral aim to provide the greatest volume of net benefits (expressed as a monetary measure of utility) for the aggregate of people affected by a decision, independent of the nature or distribution of benefits and costs.

In recent years, however, this utilitarian perspective of ecosystem service management has been evolving into an approach that better recognizes the two-way dynamic relationship between people and nature (Mace 2014, Carpenter et al. 2009). Nature and society can be seen as the two interlinked components of social-ecological systems where democratic debate and institutional regulations are recognised mechanisms to allocate environmental resources; both directly and by providing boundaries and constraints for markets to operate (Tett and Sandberg 2011). From this perspective, ecosystem services can be seen as socially and institutionally structured. More broadly, the coupled nature of social and ecological systems means they should not be seen as an independent flow from ecosystems to humans but as co-produced and culturally co-constructed (Church et al. 2014; Fish et al. 2016; Jones et al. 2016; Fischer and Eastwood 2016; Kenter 2018) by society.

While systemic ideas of ecosystem services have started to evolve and become more prominent in the ecosystem services literature, they have seen little penetration of the marine ecosystem services field. Recent economic literature on marine services has focused on scientific limitations in linking ecological functioning and services, and on the challenge of adopting stated preference valuations where knowledge on ecosystem services generated by unknown marine ecosystems is limited (Jobstvogt et al. 2014a; 2014b; Hanley et al. 2015). Moreover, this literature has little analysed the role of society, and institutions, in the production of ecosystem services (Rova and Pranovi 2017). This paper is novel in exemplifying the role of social and legislative aspects in marine ecosystem service provision, in contrast to addressing marine services only through a welfare-based economic valuation.

Society-nature interactions in marine systems have traditionally been addressed within technocratic frameworks, such as DPSIR (Luiten, 1999). Since it was proposed by OECD, the DPSIR framework has evolved. Patricio et al. (2016) evaluated more than 25 schemes produced to analyse decision making across ecosystems, but they found that there is a limited use in marine ecosystems; to fill this gap, Elliot et al. (2017) developed a new DPSIR-like concept to be more effective for marine ecosystem management. The importance of local stakeholder perspectives has been more recently recognised in the valuation (Murray et al. 2016) and quantifications of ecosystem services within broader social perspectives (Gari et al. 2015; Orchard-Webb et al. 2016). Furthermore, international surveys show that linking ecosystem services and human wellbeing, and integrating economics, natural and social sciences into ecosystem services assessments, are important research issues (Rivero and Villasante 2016; Bull et al. 2016). These contributions emphasise the need for a greater inclusion of stakeholders positions (views) into the decision-making process, as recommended by principle 11 of the Ecosystem Approach (CBD 2004).[1]

It is recognised that although the marginal approach to valuation (i.e. based on mathematical functions to assess incremental changes of the value of an environmental good or service) can provide a pathway towards the efficient management of some resources and ecosystem services (Tscherning et al. 2012), broader shared, cultural and ethical values may additionally influence the acceptability of different management decisions (Farber et al. 2006; Rehr et al.

[1] "The ecosystem approach should consider all forms of relevant information, including scientific and indigenous and local knowledge, innovations and practices"

2012; Kenter et al. 2015; Raymond and Kenter 2016) and improve fair allocations of ecosystem services amongst stakeholders (Ranger et al. 2016, Orchard-Webb et al. 2016). Decision-makers are faced with the dilemma of how to balance ecological, economic, and broader shared, social and cultural values. Moreover, addressing who is the recipient of particular benefits and costs from ecosystem services and how this inter-relates with property, legal and other rights and obligations, is an essential part of democratic governance. Monetary aggregations of individual, self-regarding preferences within CBA does not accommodate an impartial distribution of natural resources (Turner et al. 2014; Kenter et al. 2015; Irvine et al. 2016). Although distributional analysis of impacts and equity weighting can be included into CBA (Pearce et al. 2006), this raises questions of how to weight different benefits and costs and dimensions of value, generating the need for deliberation (Kenter et al. 2014b; 2015). For example, multiple "balance sheets" (Turner, 2016) can be generated to accommodate plural and collective views into decisional processes particularly to address the management of natural resources at local scale. The recognition of pluralistic ways of conceiving of and integrating plural values and non-statutory social norms through deliberative processes (Kenter 2016a; Kenter et al. 2015) takes the valuation of ecosystem services well beyond the utilitarian framework.

In this paper, we show some methodological and axiomatic issues in the monetary valuation of ecosystem services and then, building on the social-ecological systems model proposed by Tett and Sandberg (2011), we explore the limits of the choice of a pure utilitarian approach in a marine and coastal context, with examples of three different marine ecosystem services. Tett and Sandberg (2011) model, formulated to support Integrated Coastal Zone Management (ICZM), proposed decisional mechanisms within the environmental governance of natural resources as triggers of supply and demand of ecosystem services not yet described in recent ecosystem services frameworks proposed for example in the UK NEA (2011) and by IPBES (Diaz et al. 2015). We show how to synthetize the link between the Tett and Sandberg (2011) model with the McGinnis and Ostrom (2014) framework for common resource management to complement Habermas' theory of society (1987) that deals little with human-environmental relationships, to explicit the influence of social and institutional aspects on the traditional allocations of resources vaguely expressed in the UK NEA (2011) and IPBES (Diaz et al. 2015) frameworks.

2. Methodological and axiomatic issues in ecosystem services valuation

Policy-makers and regulators are placing increasing demands on economists to supply environmental values for use in policy analysis (Hanley et al. 2015). For example, EU Directives such as the revised Bathing Waters Directive (2006/7/EC) and the Marine Strategy Framework Directive (MSFD- 2008/56/EC), and the 1996 amendments to the US Safe Drinking Water Act, all depend on economic analysis to assess the costs and benefits of improving water quality. In the UK, recent conservation legislation embedded in the Marine and Coastal Access Act 2009 and the Marine Scotland Act 2010 requires economic impact analysis to assess the benefits and costs of designating new marine protected areas (MPAs). Benefits and costs of implementing such policies can be synthesized in the CBA framework to provide a ranking of more or less efficient options. Furthermore, estimation of benefit - cost ratios can identify cases where derogations can be sought if costs are extremely high compared to the social benefits (Atkins et al. 2007). Examples of studies that resulted from this policy impetus include Hanley et al. (2003) and Hynes et al. (2013), who assessed the benefits of improving coastal water quality, while Ostberg et al. (2012) and Norton and Hynes (2014) assessed the monetary benefits of achieving good environmental status (GES) in EU marine waters. Research within the UK addressed the economic benefits for designation of a network of MPAs, showing figures of nearly £17 billion in present value

(Mc Vittie and Moran, 2010). Estimates of cultural and recreational values of MPAs were undertaken as part of the UK National Ecosystem Assessment by Jobstvogt et al. (2014a) and Kenter et al. (2013). The latter estimated the present value of additional non-use benefits of marine protection between £0.7 and £1.3 billion and a benefit-cost ratio of between 1.1 and 5.8 for designating a comprehensive network of 127 new MPAs in England (depending on what implementation measures would be put in place). The importance of valuation is recognized also in those aspects of coastal management that have not yet received formal legislative support. For example, the implementation of "blue carbon" markets (Hejnowicz et al. 2015) can be facilitated by showing monetary tradeoffs between different ecosystem services and associated beneficiaries (Beaumont et al. 2014).

Two main issues are emerging in the environmental economics of resource management. The first is the reliability with which ecology and economics can calculate a utility change from a change in ecosystem services provided by the environment. The second concerns the need to understand the limits of neo-classical economic approaches in relation to their ability to inform decisions around natural resource management, and what other kinds of valuation can or should be used to complement economic information.

Concerning the first issue, we note that what is required is a quantitative understanding of linkages between ecosystem function, ecosystem services, and human well-being. In many marine cases, we do not know enough about these links. For example, we have limited and uncertain knowledge about the role of the deep-sea in terms of supporting and regulating services, and there may be significant unknown potential in terms of genetic and chemical resources (Jobstvogt et al. 2014b). In the case of energy production from marine renewables, the valuation of the resulting externalities on provisioning (fisheries), and regulating (climate change, nutrient regulation) services remains uncertain (Papathanasopoulou et al. 2015). In other cases the linkages are known to be non-linear, as for example shown for mangrove ecosystems by Barbier et al. (2008) and in some cases involving tipping points with abrupt changes in species abundance, community composition and trophic organization in the marine food web (Collie et al. 2004). Furthermore, economic valuation techniques also have significant limitations. For example, the use of stated preferences methods for valuing biodiversity, where participants are asked for their willingness to pay for a gain or willingness to accept for a loss of ecosystem services, is limited by stakeholders' lack of knowledge and experience of ecosystem functions and services (e.g. those of the deep sea), although researchers have nonetheless attempted to assess values for these unfamiliar ecosystem services in recent years (Glenn et al. 2010; Armstrong et al. 2012; Jobstvogt et al. 2014b). There have been only very limited changes in environmental economic valuation methods during the last 40 years (Hanley and Barbier 2009) to address such issues; primarily the introduction of choice experiments for natural resource valuation and the more recent development of deliberated preferences techniques (Alvarez-Farizo et al. 2007; Kenter et al. 2011; Kenter 2016b; Kenter 2017).

Concerning the second issue, even when ecological and economic uncertainty can be minimized and a robust economic valuation carried out, policy and management justified by efficiency-based economic tools can suffer from challenges to decisions when they do not address equity issues and reflect broader shared and cultural values (Irvine et al. 2016). Moreover, where national policies increasingly require technocratic tools such as CBA, at local and regional scales decision makers are more likely to benefit from evidence that is generated in a more participatory way, and which engages key stakeholder interests (Reed 2008; Ravenscroft 2010; Kenter et al. 2014a,b; Ranger et al. 2016). In some cases, the adoption of CBA on its own limits the pluralism required to address multifunctional systems with many stakeholders (Seppelt et al. 2011), and, furthermore, addresses only anthropocentric and single users' preferences when valuing ecosystem services benefits

(Burkhard et al 2014; Queiroz et al 2015). Science has a role to play in this deliberation, but new tools addressing the issues of a post normal science are required (Funtowicz and Ravetz, 1994, Ravetz 1996, Waltner-Toews et al. 2008; Ainscough et al. 2018). There is a need for an integrated natural and social science approach as part of an ongoing adaptive assessment process to understand how public and stakeholder values and preferences respond to new information and deal with more decisional power to reduce uncertainty (Funtowicz and Ravetz 1994; Milon and Scrogin 2006). Thus, a broader vision encompassing a range of tools and approaches is needed, to allow differences in stakeholders' interests to be deliberated in public and diverse views and values to interact with ecosystem governance (Fisher et al. 2015; Hattam et al. 2015). A possible way to bring in alternative institutional and social approaches is offered by the concept of social-ecological systems (Berkes and Folke, 1998). In some social-ecological models describing ecological functions to humans, integration between ecological functioning (natural component of the ecosystem) and human activities (the economic component of the anthropogenic system) has been facilitated by the implementation of the DPSIR approach (de Jong et al., 2012; Nassl and Löffler, 2015; Patricio et al., 2017). Other approaches to socio-ecological models more applicable to our study emphasize the importance of governance as the element balancing the two-way interactions between human wellbeing and ES provided by natural capital (stocks of natural assets which include geology, soil, air, water and all living things) (UK NEA 2014; Diaz et al., 2015). Governance is also seen as one of the four elements determining the action-situation of common pool resources in Ostrom (2007) and McGinnis and Ostrom(2014) (Figure 1).

Figure 1 here

Rova and Pranovi (2017) have linked Ostrom's approach explicitly to an ecosystem services conceptualization. Under this framework, the services may be either what Rova and Pranovi (2017) distinguish as 'mediated flows' (in which the resource units are themselves consumed and mediated by the need of human and built capital, for example by a fishery) or what they call 'direct flows' (in which the units are used for their ecological functions).

Our approach uses the social-ecological framework proposed by Tett and Sandberg (2011) and Tett et al. (2013), which distinguishes three ways in which society manages ecosystem services.

Figure 2 here

Figure 2 is a conceptual model that is both an ontological representation of key processes in society in relation to ecosystem services, and an epistemological claim that the analysis of such a system requires the joint efforts of economics, sociology and ecology to determine solutions that must have the property to be economically efficient, socially equitable and ecologically sustainable. The diagram shows three mechanisms that distribute services provided by natural capital: the market, top-down regulation, and collective arrangements. The market is the domain of individual interacting choices by consumers and firms. Top-down regulation seeks to harmonise societal life at a central level. Collective arrangements are the agreements and rules that groups harness to inform bottom-up collective actions. These three mechanisms correspond to three components of Habermas' model of society (Habermas 1987, figure 39). The first two mechanisms involve the institutions of media-steered subsystems, where the media are money (in the case of the market) and power (legitimately exercised by elected governments). The third set of mechanisms are those of the public and private spheres that in the Habermas' model are called *lifeworld*, i.e. the environment where ideas of social importance are mediated through cooperative action undertaken by individuals based upon mutual deliberation and argumentation. Because there is little in Habermas' theory that deals with human-environment relationships (Skollerhorn 1998; Cameron 2009), we combine the concept of the three mechanisms with Ostrom's categorization of the factors needing to be evaluated in understanding 'action situations' that involve the use of natural resources (Figure 3). The primary factors to be described are those specifying the type and functionalities of the *resource system,* the *resource units,* the (resource) *users*, renamed as *actors* by McGinniss and Ostrom (2014), and the *governance system* of the local social ecological system (Ostrom 2007, 2009). We equate *resource systems* with ecosystems including any human modifications thereof, and the *resource units* with what is providing the ecosystem service required by the *users*. Ostrom's *users* can be co-located within the two spheres of the lifeworld, while Ostrom's *governance system* (Figure 1) whose component are the Economic System, the Administrative System, and Civil Society, corresponds to the main mechanisms of our three routes for societal management of the benefits of ecosystem services (Figure 2), and to the *Economic System*, the *Administrative system*, and the *Private & Public Spheres* proposed by Habermas' model of society (1987) (Figure 3).

Figure 3 here

Finally, almost all social ecological systems (excepting that of the planet Earth as a whole) are embedded in larger scale social ecological systems, and it is thus necessary to take account of *related ecosystems* and *social, economic and political settings* (Ostrom 2007). Our three mechanisms tend to operate on different scales: the life world mechanisms are largely local, whereas the economic settings correspond to the scales of national economies which are in turn strongly influenced by globalised flows of commodities and capital, and the political (and legal) settings are in the UK's case ostensibly national, but in fact strongly influenced by trans-national agreements including, for the time being, those associated with membership of the EU. Thus, it will be necessary to take account of tiers of governance (Ostrom 2007).

3. Implementation of the methodological approach in three examples from the UK

Taking the classification of services provided by the UK National Ecosystem Assessment (UK NEA 2011; 2014) and CICES (2017)[2], we consider two regulating services (assimilative capacity from a sea-loch and coastal defense provided by managed realignment), and one provisioning service (generation of renewable energy from offshore winds and marine physical processes), whose location is shown in Figure 4. The aim is to understand and describe how utilitarian (money-steered) solutions may run up against complex social issues, and suggest how the traditional allocation of resources might be better integrated within social and institutional processes to help achieve sustainable use of marine ecosystem services.

Figure 4 here

[2] The UK NEA (2011) does not consider energy production an ecosystem services like CICES (2017), but provides the definition of assimilative capacity used in section 3.1

3.1 Assimilative capacity

The assimilative capacity is the ability of a water body to absorb a certain amount of the unintentional by-products of a human activity and to convert them in non-damaging and even beneficial products (Tett et al. 2011a). The UK National Ecosystem Assessment (UK NEA, 2014) views assimilative capacity as an indicator for the intermediate ecosystem service contributing to the removal of waste which falls into the category of regulating services. Although it is important for accounting reasons to avoid the monetary quantification of intermediate services (Turner et al. 2014), it is worthwhile to explain the indirect benefits accruing to society from assimilative capacity and in what terms economics can play a pivotal role in protecting it.

To exemplify the importance of this service, we refer to the specific case of Loch Creran (a *resource system)*[3], a sea-loch (fjord) in western Scotland (Tett and Wallis 1978; Tett 2008) (Figure 5).

Figure 5 here

This is a site where farms for salmon and bivalves, moorings and yacht anchorages, and a Special Area of Conservation (an EU protected area designation) co-exist (Buchan and McConnel 2006; Tett 2008). Therefore, amongst the obvious goods/benefits (*outcomes*) accruing to people and society from this loch are those categorized by the UK NEA (2014) as "farmed food" and "tourism and nature-watching". The benefits from the farming of fish and from services to tourism and recreation depend on two key ecosystem processes underlying assimilative capacity: the circulation of water and the burial and breakdown of organic matter

[3] In the sections 3.1, 3.2 and 3.3, the terms in the bracket and in italics refer to element of the governance for common resources as described by McGinniss and Ostrom (2014)

(Figure 6), which neutralize or export nutrients, brought into the system from the catchment and fish farms, that might give rise to eutrophication. However, if the nutrient emissions exceed a threshold, not only will there be contamination of the watercourses, but the buffers' assimilative capacity will be impaired (Fromm 2000). Under the latter circumstance, the assimilative capacity can be treated as a normal economic good showing a decreasing benefit as a function of the level of service provided.

Figure 6 here

If there were no such capacity, fish farmers would need to collect and process waste as city-dwellers are required to do. This suggests a simple method for pricing the assimilative capacity based on replacement cost of installing waste treatment at a fish farm to achieve degradation of, for a typical farm with a consented maximum of 1500 tonnes of fish, 104 tonnes of nitrogen and 25 tonnes of phosphorus per year (Tett et al. 2011b), a waste production equivalent to a city of about 10,000 people (Tett et al. 2008). Other approaches valuing welfare at the aggregate level could price this service based on stated or revealed preferences for marginal changes of recreational activities in the loch determined by water quality modifications (Hanley et al. 2003, Hynes et al. 2013), or changes in productivity in capture fisheries or shellfish aquaculture by the production function approach (Oczkowski and Nixon 2008).

The possibility to get benefits from this service depends on the regulatory and economic tools arranged by the *governance system* to reduce free riders. Although fish farms pay rent to the Crown Estate (an independent commercial business managing a real estate portfolio owned by the UK Government) for their use of the seabed as an anchorage, this rental does not explicitly include any element of paying for ecosystem services. The Scottish portfolio of the Crown Estate, including almost all fish farm leases, has recently been transferred to the oversight of the devolved Government of Scotland (Anonymous, 2017). A recent review (GVA Grimley Ltd, 2016) states rents to be 0.025 GBP per kg annual fish production and justifies these in relation to industry profits rather than use of marine resources. Because the cost of pollution is not internalized, water pollution represents an aspect of the tragedy of the commons (Hardin 1968) and it is rational to make use of “free” assimilative capacity rather than sharing the cost of farm waste treatments. The limited progress so far made with the solution of this problem is addressed by regulation, without necessarily achieving efficiency. Scottish local governments give development consent to farms subject *inter alia* to the Scottish Environment Protection Agency (SEPA) issuing a license to discharge waste under the Scottish 'Controlled Activities regulations' (Tett et al. 2015), and set an upper limit to a loch's farming capacity, based partly on the “Equilibrium Concentration Enhancement” model of Gillibrand & Turrell (1997). In other words, the estimate of capacity depends on a socially-agreed limit to the use of the watershed, by reducing the intensity of aquacultural production and of agricultural inputs (i.e. organic production, tuning timing of fertilization and ploughing, etc. - Schoumans et al. 2014), or by promoting integrated multi-trophic aquaculture where shellfish or seaweed absorb some of the salmon nutrients (Tett et al. 2011b).

In the case of Loch Creran, limits imposed by SEPA have been respected and no environmental externalities have been emerged so far. However, if these regulations failed, assimilative capacity would be reduced (i.e. become, in economic terms, a scarce good) and then perceived as a valuable resource (price higher than zero). Economics could intervene in this case allowing for users of this service to bargain the reduction of the harmful activities against monetary compensation so as to achieve the solution that provides the highest net social benefits to the parties overall, including the public (Coase 1960). Market approaches to environmental protection such as water quality trading have been adopted in the U.S., Canada, Australia and New Zealand (Selman et al. 2009). However, efficacy may be reduced by the likely high number of stakeholders involved and consequent transaction costs (Coase 1960). This makes the case for an alternative approach, where the state intervenes when pollution becomes a social concern, based on charges to remove the market failure of unaddressed pollution in economic transaction of marketed goods (Pigou 1932), as adopted with success in the Netherlands (Elkins 1999). Both approaches are meant to achieve environmental goals at a lower cost to society than a regulatory solution (UNEP 2010), reaching the point that makes the polluter indifferent between further treatment and discharge of pollutants. In other words, each polluter will be reducing pollution up to the point that the marginal cost of pollution to the society will equalize his private benefits from polluting.

Such economic mechanisms can succeed only under well-established institutions that set who has the right to use the service (fish farm polluter, agriculturist, etc.) and who has the right to be compensated. In the example of Loch Creran, an alternative approach to regulation is not needed because none of the three types of signals (economic, social, institutional) described in Figure 2 have emerged so far. Economic valuation of assimilative capacity requires perception of the scarcity of this ecosystem service; social analysis of assimilative capacity requires the emergence of a sense of health danger amongst stakeholders; and institutional intervention requires the definition of property rights between actors before discussing the optimal level of externalities and marginal rate of economic exchange between the parties.

3.2 Coastal defences

To counteract the effects of coastal flooding, stronger hard defences are often demanded by citizens, whereas public bodies are concerned with the high construction and maintenance costs of adequate sea-walls. Furthermore, such defences may also impact on the integrity of coastal habitats (French 2001) that if kept in place can provide ecosystem services, including natural hazard protection (absorbing flood and storm energy), and can generate economic benefits in terms of avoided damages to coastal capital assets as well as satisfying well-being needs arising from broader social and cultural values (Figure 7).

Figure 7 here

Natural defences, if trapped between a sea-wall and the rising sea, are subject to accelerate erosive processes and in turn unprotected sea-walls become increasingly exposed to high energy waves. An alternative to hard defences is managed re-alignment of coastal habitats (*resource system*). The sea is allowed to flood low value land and convert it to natural coastal habitats, which slowly adjust to rising sea level: beaches maintain natural profiles and salt marshes dampen wave energy and trap silt, protecting from storm and wave actions (Defra and EA 2005; Feagin et al. 2008; Koch et al. 2009). In addition, coastal saltmarshes provide other benefits in terms of regulating services (*outcome*) such as carbon sequestration (Cannell et al. 1999, Choi & Wang 2004, Beaumont et al. 2014), functioning services such as nutrient cycling (Howard et al. 2014), sheltered nursery sites for fish (Laffaille et al. 2000), and high tide refuges for birds, contributing to fulfil nature conservation objectives such as those associated with the creation of the EU Natura 2000 network (Doody 2008). Multiple ecosystem services such as healthy climate (via carbon sequestration and storage), food provision (via fisheries production in nursery grounds), and nature recreation (nature watching and enjoyment) specifically provided by managed realignment schemes are assessed by Luisetti et al (2011; 2014).

Several federal and national administrations provide public support to local projects if they prove to generate benefits to the public and cover capital and operating costs. This is the case in the USA (NOAA 2006) that in deciding how to allocate federal money to single states for beach nourishment programmes requires the adoption of CBA for projects that receive public funding. Similar considerations apply to the UK where CBA is required to assess the benefits of policies (HM Treasury, 2003). Spending on flood defences has to be justified by benefits such as the protection of property of a certain total value; when the value preserved is too little, houses and farms may be abandoned to the rising waters. When the case for adopting alternative strategies to hard defence of coastal capital assets is strong according to an environmental CBA (taking into account externalities), managed realignment can be considered a viable solution. The UK Natural Capital Committee (NCC 2015) action plan for saltmarsh protection estimates a benefit-cost ratio of 2 - 3:1 in flood reduction and habitat gain from managed realignment, even without taking into account possible additional and long term gains provided by carbon sequestration or other unpriced regulating services beyond flood protection.

In these cases, the coastal defence service is valued at what it would have cost to build or maintain sea-walls, less the value of sacrificed land, plus the value of services provided by the new natural habitats. Shepherd et al. (2007) and Andrews et al. (2006) estimated that managed realignment projects would pay off after 25 years for schemes applied to the Humber estuary (northeast England) and after 50-100 years for the Blackwater estuary (southeast England), due in part to the value of the additional ecosystem services created by the re-alignment. Examples of CBA where net present values (NPV) of managed schemes are compared to NPV of keeping the defence (*status quo*) are provided by Turner et al. (2007) and Luisetti et al. (2011) for the Humber and Blackwater estuaries, respectively, under different policy scenarios, discount rates and lifespan of the schemes. For the Blackwater, NPVs were between -£24m and £37m, casting doubt on the viability of a realignment. The Humber case, however, showed only positive NPVs, between £70m and £300m. The results for the two examples highlight that the values are sensitive to the scale of the realignment scheme and the local topography, and thus a general conclusion cannot be provided (Luisetti et al. 2011; 2014).

The Steart case (southwest England) is more complex. The low-lying land of the Somerset Levels is at risk of flooding from both the sea and rivers. A managed realignment, to better protect against the risk of marine incursions and at the same time to compensate for habitat losses to hard defences elsewhere in the region, had been planned since 2002 and had been evaluated as showing a net benefit to ecosystem services (da Silva et al. 2014). However, a few months before the scheme was completed in September 2014, river-derived flooding inundated large parts of the Levels. The causes of this flooding were disputed, ranging from unusually heavy rainfall on already-saturated catchments (Anonymous 2014) to failure to dredge the river channels, and the flood events led substantial popular doubt and opposition in relation to the re-alignment works, backed up by hostile reporting in some newspapers and issues raised in debate in the UK Parliament[4]. Although the Steart scheme was intensively researched before it was executed, there remain at the time of writing competing expert and popular narratives about flood prevention in the Levels that CBA has been unable to reconcile.

This illustrates that environmental CBA is challenged where managed realignment policies encounter political difficulties: in many cases, there is distrust among local communities about government intervention and plans for coastal adaptation measures (Luisetti et al. 2011). Abandoning land and property to the sea is considered as dereliction of a state's core duty to protect its citizens. In the case of the Steart, opponents claimed that birds were put before people and that the £20m spent on the re-alignment should have gone to hard flood

[4] HANSARD Commons 22 Jan 2014 vol 574 columns 124WH – 134 WQH, speech by Mr Ian Liddell-Grainger MP in debate on flooding

defences instead of a "bird sanctuary"[5]. This opinion, steered by a sense of insecurity following widespread river-basin flooding, expresses different value priorities than reflected in the CBA, but also there is uncertainty of evidence that managed realignment will deliver comparable ecosystem functionality to natural sites at different points in time (Morris 2013). Luisetti et al. (2011) conclude that managed realignment, when there are potential flood risks to a significant number of people and assets, provides a context for complex decision-making in which CBA does not necessarily provide decisive information on trade-offs.

[5] www.theguardian.com/ environment/2014/sep/08/20m-pound-salt-marsh-somerset-wildlife-habitat-ghts-sea-erosion.

3.3 Marine renewables

The marine environment (*resource system*) can represent a source of energy potentially meeting higher energetic demand (Astariz and Iglesias 2015). Offshore renewables are likely to play an important role in a suite of technologies (Pelc and Fujita 2002), with wind, tidal and wave energies (*resource*) a viable part of energy supply (Voke et al. 2013). The governance system for offshore developments is not yet well framed in many countries (Wright 2015), but it is commonly accepted that it requires (i) environmental impact assessment (EIA) for reducing damages, (ii) conflicts minimisation between uses and users (Scott et al. 2016; Graziano et al. 2017) to provide social and economic net benefits, and (iii) a flexible regulatory framework (Wright 2015) that neither impairs the needs of developers nor the wishes of civil society for an healthy environment. Considering the relatively young age of marine renewable energy technologies, a gap in the knowledge of the information-chain between the final service (energy) and effects on human wellbeing is evident in the literature. An overview of environmental impacts on marine habitats and changes in ecosystem services are provided by Papathanasopoulou et al. (2015) for nuclear, oil and gas and offshore wind farm (OWF) industries. Some studies that focussed on OWF impacts and marine renewables are categorised according to the ecosystem services framework in Table 1. Overall, they show contradicting results on the likely impacts on provisioning and cultural services, and thus far little evidence in terms of potential impacts on regulating services.

Table 1 here

In terms of the latter, the observed increase in mussel abundance at OWF is likely to increase the capacity of the system to remediate waste as well as increasing food availability for other species (supporting service) (Wilhelmsson and Malm 2008). As regards provisioning services, there is a general lack of studies which investigate impacts of OWF on commercial fishery species. OWFs may have positive effects on commercial fish and shellfish stocks through artificial reef effects (Wilhelmsson et al. 2006). It is found that the temporary closure of European lobster fishing ground due to OWF construction can benefit the increase in abundance and size (Roach et al, 2018). Mussels, brown crab, cod, pouting, eel and sole have been found to be more abundant near offshore turbines (Hooper and Austen 2014), but increase in abundance or biomass of fish and benthic species is not universal across all sites (Ashley et al. 2014) and does not automatically lead to an increase in catch rates within the fishery (Hooper et al. 2015).

Studies on the perceived impacts of marine renewables on cultural ecosystem services are also emerging. Ladenburgh (2009), reviewing some studies on the preferences for OWF in relation to onshore development, showed that these preferences seem to be dependent on the specific place of location offshore. However, aesthetic impacts can be significantly reduced by locating wind farms at large distances from the coast (out of sight), a popular solution evidenced by a positive willingness to pay by the public (Ladenburgh and Lutzeyer 2012). Similar results are found by Voke et al. (2013) who assessed by using contingent valuation and travel cost methods that the potential impact of renewable energy generation on tourist visiting Pembrokeshire (UK), would put off visiting a limited number (3.5% of the sample) due to a visible marine energy development. This suggests that the potential impacts on the recreational activities in coastal zones affected by the development of newer forms of marine renewable energy (tidal currents and waves) could be less adverse than those shown for OWF by Ladenburgh (2009).

This classification of benefits and costs arising from marine renewables helps identify impacts and may facilitate a more transparent adoption of environmental CBA and EIA procedures (Wright 2015). However, ecological knowledge gaps about changes in marine functioning following marine renewables deployment weaken the possibility to value services by the production function approach and stated preference methods. For example, in a choice experiment for the establishment of an MPA on the Dogger Bank (North Sea), while people showed willingness to pay for environmental improvement, about 25% of the sample stated that they did not have enough knowledge about the issues to make a choice (Borger et al. 2014). The difficulty and uncertainty in quantifying intangible costs and benefits that do not have a market (for example changes in conservation values of bird and marine mammals wildlife from the implementation of renewables) cause limitations in the use of CBA as a decision support tool, providing a lower confidence in the usability of the indicators of the economic acceptability of the project (Wright 2014). In addition, we must mention that the difficulties of addressing in a single monetary indicator the pluralities of views on renewables and the aversion of regulators for projects they perceive as risky (Neumann 2009) can lead to conflicts between stakeholders.

A case study exemplifying these complexities is that of marine wind-farm developments to the east of Scotland. Four large interrelated developments totalling 0.5GW supposed to generate between £314m and £1.2bn for the Scottish economy and to create more than 500 jobs during construction and more than 100 permanent jobs once operational were at stake[6]. Licences granted by the Scottish Government to allow these developments were overturned in 2016 by a judgement in the Scottish Court of Session.[7] Publicly available documents from one of these studies, that of the Inch Cape Wind farm,[8] show that there was no explicit CBA

[6] 'Judgement threatens multi-billion pound Tay and Forth wind arrays'. Graham Huband: Dundee Courier July 19 2016. The Courier.co.uk. Three development companies were involved: Inch Cape Offshore Limited, owned by Repsol Nuevas Energías UK (51 per cent) and EDP Renewables UK (49 per cent); Seagreen Wind Energy Limited, a 50/50 joint venture partnership between SSE (Scottish and Southern Energy plc) and Fluor Limited, the UK operating arm of Fluor Corporation; Geart na Gaoithe Offshore Wind Limited, owned by Mainstream Renewable Power.

[7] Scotland Court of Session, Outer House: [2016] CSOH 103. Petition of The Royal Society for the Protection of Birds for Judicial review

[8] EIA and other documents can be found at: www.inchcapewind.com

for wind farm and impacts (externalities) to support the Government decision. However, the farm's Environmental Impact Assessment, required by law, considered these impacts and how to ameliorate them. It included an account of the benefits in jobs and extra income in the 'Economic Study Area' in eastern Scotland, and more widely in terms of reduced emissions of greenhouse gases.[9]

Development consents were granted by the Scottish Government in 2014, and were challenged by the (UK) Royal Society for the Protection of Birds (RSPB). The 2016 judge summarised the RSPB concerns for Atlantic puffin (*Fratercula arctica*), northern gannet (*Morus bassanus*) and black-legged kittiwake (*Rissa tridactyla*) as risk of collision with turbines, displacement from foraging areas and barrier effects such as foraging flights to and from breeding colonies. The latter two risks may entail extra energy costs and consequences for body mass, adult survival, nest attendance and chick provisioning (Scotland Court of Session, Outer House: [2016] CSOH 103. Petition of The Royal Society for the Protection of Birds for Judicial review).

The judgement is particularly interesting because it was decided in part by scientific evidence. In particular, the judge accepted RSPB arguments that the present state of scientific knowledge and art (in population modelling) was insufficient to show that the farm would not impact adversely on protected populations of seabirds. Under the European Union Birds Directive (72/49/EEC replaced by 2009/147/EC), which protects the birds, and the Habitats Directive (92/43/EEC), which protects their habitats insofar as they are designated, the precautionary principle applies, guaranteeing protection excepting imperative reasons of overriding public interest (e.g. human health or public safety). The consequence is that potential damage to bird populations cannot be traded off against benefits in reductions of

[9] Chapters 8 and 22 of the EIA of 2013.

greenhouse gas emissions. The judgement also found that there had been procedural flaws in the way the licencing process had consulted stakeholders.

The Scottish Government appealed this decision, and it was overturned in 2017 by the most senior judge of the Supreme Courts of Scotland.[10] He concluded that the Government followed the law in giving consent and that the 2016 judge should not have tried to evaluate the scientific evidence. Clearly, the senior judge was trying to keep the law out of science and of decision-making based on scientific evidence. Yet this simply moves the locus of decision from one hierarchy (the law) to another (the government and its scientific advisors). Of the two alternative methods shown in Figure 2 for regulating the use of marine ecosystem services, economic valuation of ecosystem services and environmental CBA of the project seems not to have been used in any explicit fashion, and deliberative processes were largely restricted to those between Government and Statutory Consultees, of which those relevant here were the public bodies Scottish Natural Heritage and the UK Joint Nature Conservation Committee.[11] It is possible that more ex-ante stakeholder deliberation, as recommended by Scott et al. (2016), might have led to a different wind-farm configuration allowing better recognition of non-utilitarian values and reduced the risk of lengthy and costly legal proceedings.

[10] Scotland Court of Session, Inner House: [2017] CSIH 31. Reclaiming Motions in the Petitions of The Royal Society for the Protection of Birds against the Scottish Ministers etc.

[11] Paragraphs 25, 38 of the 2016 judgement.

4 Discussion

Although there is no doubt that human wellbeing depends on natural capital and ecosystem services (MEA 2005), there has been legitimate debate as to whether utilitarian or other types of ethical orientations might best sustain these services (Fisher et al. 2015). Large-scale initiatives have been implemented to explore relationships between ecosystem functioning, services and wellbeing and to develop mechanisms to recognise, demonstrate and capture values in decision making (MEA 2005; TEEB 2010; UK NEA 2011; 2014; IPBES -Diaz et al., 2015). While the approaches taken have so far largely reflected the notion of nature for people (Mace et al., 2014), there is a clear shift towards more integrated perspectives that consider nature and people (Mace et al. 2014) as a more inclusive relationship, and emphasise the importance of local inputs (views) to increase subsidiarity in the governance and to empower actions at the scale where they can weave a canvas of resilient interactions between human society and natural environment (Carpenter et al. 2009; Mace et al. 2014; Kenter, 2016a; Jacobs et al. 2016).

In this discussion, we will (i) appraise the extent to which mainstream economic theory and methods could be practically useful in managing human-environment interactions in our case studies, (ii) provide an interpretation of the ecosystem services framework within an institutional economic context, and (iii) reflect on the opportunities for expanding the horizon of economic valuation of marine ecosystem services by using pluralistic deliberation to democratise decision making.

4.1 The role of economics

Mainstream economics emphasizes the importance of valuing via stated or revealed preferences those ecosystem services that are typically undervalued in economic decision-making because they are provided free of charge as typical public goods (Turner and Daily 2008) and because markets fail to adequately signal their true value (Ribaudo et al. 2010). This ‘undervaluing’ is often considered a key factor in the decline, degradation, and in some cases irreversible loss of biodiversity (Ribaudo et al. 2010). For these reasons, recent studies have called for methods for the valuation of ecosystem services and biodiversity, and for better understanding of their reciprocal relationships (Bennet et al. 2009) that might ultimately result in commodification of nature. Although the trend to commodification has led to protest from scientists, conservationists and political ecologists (Gomez-Baggethun and Ruiz-Perez 2011; McCauley 2006; Turnhout et al. 2013; Matulis 2014; Spash 2015), others have argued that a degree of commodification is necessary to embed protection of services in a market context (TEEB 2010; UK NEA 2011) and to facilitate long-term behavioural changes towards nature conservation (Burton and Schwartz 2013). Outside the market context, the monetary valuation of nature benefits can be used to inform trade-offs amongst development and conservation policies, although only a few successful instances of this have been documented so far, and in practice, policy makers rarely commission these for improving planning (Ruckelshaus et al. 2015; Laurans et al. 2013).

Of our three examples, the economic valuation of ecosystem services and CBA were used only in one (Steart); it would seem to have potential in another (AC), and was explicitly disallowed by one interpretation of the law in the third (wind-farm).

In relation to our first example, it seems possible that economic valuation could work in harmony with social norms and institutional rules when assessing social benefits of assimilative capacity. Currently assimilative capacity is an unacknowledged common good that generates *free riders* because of the false view that it is less costly to share the consequence of pollution rather than to internalise (e.g. privatise) the costs of treating a discharge. When assimilative capacity is not perceived as scarce, it does not generate conflicts between users, as in our example. If it becomes scarce, or is perceived as scarce, regulatory or economic mechanisms can be arranged to reduce pollution by levying charges underpinned by the monetary valuation of the externality (Selman and Greenhalgh, 2009). Different considerations apply to the benefits provided by natural coastal defence, our second example. In the case of the Steart re-alignment, CBA allowed a net positive value to be identified for the scheme. Although there were gaps in scientific knowledge (da Silva et al. 2014) and negative public reactions from the perceptions of the causes of severe inland flooding as well as agitation by the members of parliament representing the area, these did not prevent the top-down implementation of the scheme.[12] In this example, there were no overwhelming technical difficulties, for example in eliciting stakeholders' preferences, that might compromise economic valuation. Nevertheless, even when the CBA is technically feasible, as in this case, it did not provide a "heuristic aid" in the decision making process (Turner 2007) because of limitations in engaging or hearing third parties (local and lesser voices). Moreover, the difficulties to take account of a plurality of views can generate distortions in the distribution of benefits between social classes and a consequent misleading interpretation on the welfare assessed by CBA for society as a whole (Turner et al. 2007).

Finally, in the wind farm example, the decision by Government to licence development seems to have been based on policy grounds (helping to reduce greenhouse gas emissions) and perceived economic benefits, without an explicit CBA and without full monetization of ecosystem services. Although in the view of the 2016 judge, societal benefits could not be traded off against potential loss of seabirds, the 2017 judge concluded that Government simply needed to have followed the procedure laid down by the law, and that it had done this. Societal concerns emerged in the legal challenge from the RSPB, in this respect representing

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[12] Hansard Commons , 22 Jan 2014, vol 574, cols 124WH-134WH

a part of society lying mostly outside the parts of Scotland that might benefit from wind-farm-generated employment. At the least, this imperfect representation of societal concerns, and the argument within the legal hearing of the scientific knowledge gaps, has delayed the wind-farm development by three years or more. However, a greater role for economics would unlikely have helped to resolve the conundrum more effectively; rather, earlier stakeholder participation and enhanced deliberation may have allowed different interests and values to have been weighed and balanced sooner without resolving to a complex legal route.

4.2 Ecosystem services and institutional analysis

Ostrom's social ecological systems (Ostrom 2009; McGinnis and Ostrom

2014) describes how ''inputs are transformed by the actions of multiple actors into outcomes'' (McGinnis and Ostrom 2014, p. 34), while Rova and Pranovi (2017) and Nassl and Löffler (2015) focus more explicitly on analyzing the causal linkages between the system variables, explaining how ecosystem services ('*resource units'*) emerge from ecosystem structures and processes ('*resource system'*). In this study, we broaden the previous considerations on the governance of ecosystem services looking at how the institutional analysis (explaining the functions of a wide range of institutions such as government, the law, markets, etc.) would describe resource management based on choices at citizens' level (public and private spheres), to complement decisions undertaken by markets (Slavikova 2013) and government, and to individuate weaknesses in natural resource governance (Ostrom 2007; Ostrom 2009). Looking at the assimilative capacity example, the pure market approach can be used to obtain efficient solutions, if conflicts from the perceived scarcity of the resources arise, by designing arrangements including regulations, economic tools (taxes, subsidies) and creations of new markets. These rules can be figured out as hierarchical arrangements, involving a chain downwards from rulers to ruled that is underpinned by law and power, and a chain upwards from ruled to rulers (Habermas 1996) that depends on the perceived legitimacy in the allocation of rights to polluters or polluted in "consuming" the assimilative capacity. In the example of coastal defense, the CBA and market level was supported by the downwards chain from government but opposed by the upwards chain originating in citizens' wish to protect land and property from flooding. In the example of wind farms, the disagreements centered on conflicts between government agencies, developers and civil society owing to limited scientific information on offshore wind farms impacts. This distrust reveals the limited inclusivity of voices from the bottom (Young 2002; Ostrom 1990; Ostrom 2007), usually associated with decisions debated in local fora. This is evidenced by the difficulties of EIA in assisting local decision making in context of scientific uncertainties (Huertas-Oliveras and Norris 2008), and in supporting a broader democratic perspective to balance and reduce tension between opposing strategies such as the precautionary (reject any plan) and the risky approach (accept the plan even at high environmental costs) (Renewable UK 2011).

Looking at these three ecosystem services under the lens of the framework proposed in Figure 3, it is evident how the governance represented by the *Economic System*, the *Administrative system*, and the *Private & Public Spheres,* external to the institutional order of the lifeworld where communication frames the collective actions, is a limiting factor for the ability of stakeholders to engage in collective arrangements. R*esource systems* as indicated in Figure 1, or *Environment* in Figure 3, in the three examples require a number of relevant actors to influence the costs of cooperative actions (transaction costs). Moreover, a major factor affecting *users* (stakeholders of lifeworld in Figure 3) consists in the limited predictability of the dynamics of the *environment*, which led to failure in *resource* *governance* as uncertainty in the consenting process for deploying marine renewables (*Administrative system* in Figure 3); and, flaws in social capital construction, including lack of trust in public protection of private property in the *Private & Public Spheres*, which appeared to result in community aversion towards soft sea-defence schemes.

4.3 Implications of results for economic valuation, use of CBA and democratic decision making

When valuation is not severely restricted by the difficulties of using monetary metrics, CBA can play an important role as a “decision rule”, operating as one of multiple components of policy analysis (Turner 2007). However, the acceptability of the net present value of a project as a decisive element for decision making is not obvious. Although it is possible to find cases in the literature where economic valuation and CBA proved useful for management (e.g. Turner et al. 2007; Luisetti et al. 2011; 2014; Martino and Amos 2015; McDonald et al 2017), there are many others where social concerns were stronger, undermining CBA applicability (e.g. da Silva et al 2014; Roca and Villares 2012; Myatt et al 2003; Myatt-Bell et al 2002; Irvine et al. 2016).

Overall, CBA seems to have been more useful in coastal defence strategies than in managing the development of offshore renewable energy. This may have been because of the better scientific knowledge of the ecosystem functions-ES interface in coastal and intertidal habitats, and the possibility of using stated preference methods to value services provided by relatively familiar environments (Hanley et al. 2015). Conversely, complexities are emerging when people are asked to state preferences for goods or services provided by unknown environments as the deep sea (Glen et al., 2010; Armstrong et al., 2012; Jobstvogt et al., 2014b), making difficult to trade-off marketed and non-marketed goods.

Our examples show some key constraints to application of neoclassical valuation and CBA due to a lack of knowledge, about links between ecosystem functions, ecosystem services, and human well-being, among: 1) scientists working at the interface between ecology and economics to provide reliable economic valuation; and 2) users who do not receive sufficient, clear and balanced information of the services provided by ecosystems. Issues of limited information occur especially when science is not able to communicate clearly benefits arising from those services that appear as a mental construct rather than tangible goods (Barbier 2007; Turner et al. 2007; Luisetti et al. 2011). Furthermore, at least three problematic assumptions of neoclassical valuation arise (Kenter et al. 2015): 1) the idea of the "rational agent" with preferences characterised as independent and exogenous (Bowles and Gintis, 2000); 2) that all values can be expressed as preferences; and 3) that social welfare is a cardinal measure of the sum of the preferences of independent individuals (Parks and Gowdy, 2013). As regards the first and second point, behavioral literature recognises that values need not be preformed (i.e. existing as *a priori* ideas) but constructed through formal or informal deliberation (Shapansky et al. 2003; Kenter et al. 2016b,d) to shape decisions under societal perspectives rather than that of individual consumers (Lienhoop et al. 2015; Martinez-Espineira 2007; Orchard-Webb et al. 2016). As regards the third point, Arrow (1950) showed that there is "no logically infallible way to aggregate the preferences of diverse individuals" (cited by Feldman 1987: 894), therefore CBA cannot provide a consistent ranking of policy alternatives (Parks and Gowdy 2013).

It is important to recognise that these limitations are not necessarily problematic; rather, they become so in cases that are complex, contested and with many stakeholders that harbour conflicting interests (Kenter 2016a). In these cases, there is increased likelihood that peoples' values and behaviour will depart from the rational actor model and that commensuration of pluralistic values and aggregation through typical (Bergon-Samuelson) social welfare functions will raise moral and justice concerns. Thus, limits to the feasibility and legitimacy of pricing services and ranking projects by CBA are empirical (uncertainty about ES-welfare linkages), theoretical (assumptions of welfare economics are not always tenable) and political (perceived legitimacy is questioned). As summarized in Table 2 for the three examples, signals coming from civil society (shared perspectives and values, and institutional norms) may reduce the scope for economic valuation and CBA application, raising uncertainties on its applicability because of difficulties in addressing distributional equity (Pearce et al. 2006; Turner 2007; Turner et al. 2016).

Table 2 here

The efficient use of natural resources as a mechanism for improving human wellbeing according to a utilitarian view of nature can drive narrowly rational economic decisions into the realm of wicked problems where these apparently rational solutions run into complex social issues revolving around pluralistic values related to both process and outcomes. In turn, this can make the existing social-ecological system unstable (Folke 2006). Laws and customary norms may clash as exemplified by the conflicts between decisions on flood defences strategies, based on technocratic CBA, and citizens' demands to receive protection for their properties from the state; and by the judicial review of offshore wind farm consent, which was overruled by the application of the precautionary approach applied to European protected areas. These two examples make evident the policy challenges of reconciling contested value realms and achieving consensus to create more inclusiveness that encompasses a greater social diversity and power (Fabinyi et al. 2014). Thus, there is the need to move from what might be called algorithmic allocation of preferences by market or law, to a broader political debate amongst citizens by which their conflicting interests are reconciled (Sabatier 1998; Kenter et al. 2015; 2016b).

This suggests using CBA differently in the political economy domain: 1) by reducing the scope of CBA as a tool addressing rational solutions (Randall 2002) especially in those contexts where broader ethical/moral imperatives and social justice issues are of central importance; and 2) by switching to a different perspective that considers CBA as an element supporting, but not steering, decisions (Turner 2007; 2016; Turner et al. 2014). This may also involve addressing the way economics weights benefits and costs, for example by including deliberation on how they should be aggregated (Kenter et al. 2016c), and complementing economics with other forms of valuation to depict a more comprehensive vision of what is needed to ensure sustainability in contexts such as energy developments and flood protection. For example, Orchard-Webb et al. (2016) exemplify a novel approach, Deliberative Democratic Monetary Valuation (DDMV; also see Kenter 2017), where a group of local stakeholders expressed values through several interpretive and deliberative tools, then evaluated different policy options using conceptual systems modelling, visioning and multi-criteria analysis, before directly negotiating values for policies related to the marine environment. Ranger et al. (2016) demonstrated an approach where qualitative video interviews with a wide range of stakeholders were used to elicit plural values, which then fed into deliberative multi-criteria workshops where policy options were voted on. Reed and Kenter (2014) and Reed et al (2017) highlighted a study where DDMV was used to negotiate conflicts between stakeholders and directly establish fair prices for payments for ecosystem services in a peatland management context. These examples illustrate the need for deliberative processes to help form shared values between different interests and reduce conflict. A range of key factors needs to be considered that influence this process of value formation (Kenter et al. 2016d), including for example education and prior experience to process design and facilitation that can help manage power dynamics and actively support and enable participants in the translation of transcendental values, our overarching principles and life goals (Raymond and Kenter 2016), into more specific contextual values and value indicators.

Deliberation is not just an important mechanism for value elicitation and formation, but also for value aggregation. When pluralistic values are recognised to be incommensurable, it can be left to formal political and legal processes to aggregate information, but this is challenged by significant institutional barriers (Turnpenny et al. 2014), and also does not account for people's meta-values in relation to value aggregation (Kenter et al. 2016a; 2016b). To reconcile the different approaches at valuing and overcome the restrictions on applicability of CBA, Turner (2016) proposes that a modified form of CBA could be part of a *triple balance sheet* approach, a societal decision support system (Figure 8) that goes beyond the framework of economic efficiency to take account of shared values, equity, justice, and plurality of views. The framework suggests that in simple and non-contested contexts, a single balance sheet consisting of a modified version of CBA is adequate. When there is increased complexity and contestation, two further sheets can be added which address social impact assessment and broader shared and cultural values. Values are aggregated and value conflicts negotiated through deliberation. This provides a basis for collective arrangements for overcoming barriers, addressing social challenges and seizing opportunities in a "pragmatic" perspective where individuals and groups work towards agreement from initially contesting positions (Bromley 2004). This approach through its greater potential for Communicative Action (Habermas 1984; 1987), may be more fruitful for the understanding, improvement and implementation of environmental policies than the rational choice theory that holds that "correct" positions can be formulated by *pre-determined* (innate) preferences rather than being constructed by a deliberated process (Scott, 2000).

Figure 8 here

5 Conclusions

We have discussed the complexities of societal management of three ecosystem services. In the case of assimilative capacity for nutrients and waste in a sea-loch, there is scope for improving efficiency by imposing a charge for the use of this public good, in a way that harmonises with existing governance. In the case of coastal defence, the expert monetisation of the service and prioritisation of decisions under the CBA framework has conflicted with the way many local people think the state should protect life and property. In the case of offshore wind farms, the conflict represents two different perspectives on value, which were not reconciled in the planning process, leading to its ultimate breakdown through legal action; a more bottom-up, participatory approach that focused on reconciling different interests and values from the outset (rather than maximising net benefits) could conceivably have led to a different outcome.

Provision of ecosystem services requires sustainable management of the natural capital underpinning these services, but equally the role of human, social and built capitals needs to be considered when assessing how these services and their benefits to human wellbeing are generated (Haines-Young and Potschin 2010; Costanza et al. 2014; 2017). Study of the complex interaction of these forms of capital can build on the foundation of institutional analysis (McGinnis and Ostrom 2014). Under the consideration that many ecosystem services (mainly the regulating services) are public goods, poorly recognised within the market domain, institutional analysis is a necessary complement to economic analysis to individuate weaknesses in natural resource governance (Ostrom 2007; Ostrom 2009). The complexities in decision-making shows that the strategies relying exclusively on removing market failure do not produce satisfying results in resource management unless they also promote participation in decision making by local stakeholders (Dietz et al. 2003). As we have illustrated, the requirements for managing ecosystem services are multiple and need not only economic information, but also capacity to deal with social conflicts, and flexibility to allow governance to better adapt via broader participation. Therefore, a mix of institutional types promoted by well-designed deliberation involving scientists, resource users and citizens is needed to integrate evidence and values, to bridge inevitable conflicts, to enable social learning and to implement a diversity of adaptive governance strategies that can effectively and sustainably oversee natural resources.

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Figure 1: A multi-tier framework for analyzing a Social-Ecological System', redrawn with slight modifications from Ostrom (2007)

Figure 2: A social-ecological system (Tett & Sandberg, 2011) showing the three routes by which ecosystem services can be allocated within human societies. A feedback loop from society to ecosystem considered is not shown, but we recognize that natural events and human activities are the drivers of pressures on the state of the ecosystem; changes in ecosystem services impact on society and bring about a management response. Some of the allocation mechanisms include feed-back loops (e.g. the response of price changes on demand for a particular service; see UK NEA, 2014 and Diaz et al., 2015 for a specific description of a circular pattern between ES provision and institutional response).

Figure 3. A social ecological system linking the Habermas view of model of society (1987) with the social-ecological system provided by Ostrom (2007) and shown in Figure 1. Ostrom's *users* are to be found in within the two spheres of the lifeworld, constrained there by societal norms, and within the two media-steered major subsystems of organised society, where they are constrained by the relevant medium.

Figure 4: map of the case studies. On the left a map of the UK. On the right a zoom in the sites where example are borrowed. Loch Creran (figure 4.1) is a fjord in the Argyll region of Scotland and is considered for the social-economic analysis of assimilative capacity; the Somerset Levels (figure 4.2) are a coastal plain and wetland area in south west England and are considered here to discuss the case of coastal managed realignment; the Firth of Forth (figure 4.3) is the estuary of the River Forth and several other Scottish rivers and is considered for the analysis of the offshore wind farm case.

Figure 5: Assimilative Capacity of a water body for an anthropogenic waste
Seawater circulates through the example water body, loch Creran. The concentration of anthropogenic waste added to the loch (for example by a fish-farm) depends on the rate of loading with the waste, the rate of its loss to local removal processes, and the rate of loss to

the water circulation. Other things being equal, the concentration increases with load. Regulators set a maximum allowable concentration (an Environmental Quality Standard, EQS) of the waste. Assimilative capacity is the excess of this EQS over the background concentration of the waste substance in the seawater that is flushing the loch. For further details of loch Creran's circulation see Tett (2008) and Tett et al. (2011a).

Figure 6: Assimilative Capacity (AC) is a bio-physical measure of the ability of an ecosystem to absorb anthropogenic inputs of substances without damaging the health of the ecosystem or its ability to provide goods and services (Tett et al., 2011). In the present case, the substances are organic waste, plant nutrients, and fish-treatment medicines. AC relies on the proper functioning of ecosystem processes including, in the case study, dilution of loch contents by external sea-water, cycling of nutrients, and decomposition of organic matter. AC is not infinite and is bounded by the intrinsic limits of the environment. If overcome, this function is lost and provides a reduction of water quality, which generates disutility to the end users.

Figure 7: Linear cascade between functioning ecosystem services and benefits for the coastal defence case study where the production of new coastal habitat provide control of waves regime and facilitate the protection of the inwards capital assets and uses of land such as agriculture or urban/industrial expansion. Source: adapted from Luisetti et al., 2011

Figure 8: The Triple Balance Sheet approach (simplified from Turner et al., 2016). `Tame' problems relating to the use of ecosystem services should be soluble by straightforward economic analysis as in the left-hand column. More complex and `wicked' problems (Jentoft & Chuenpagdee, 2009) require the additional columns of analysis

Table 1: Summary of some recent studies on environmental impacts of renewables on the marine environment arranged under the logic of the ecosystem service terminology (according to UK NEA (2011) and CICES (2016)

Ecosystem service	Issues	Comments	Authors
Provisioning services	Commercial fishery	Higher abundance of commercial fish and shellfish through artificial reef effects, but not recognised universally;	Wilhelmsson et al. (2006);
		Species richness and diversity reduced around the turbines;	Hooper and Austen (2014);
		No direct effects on catch rates within the fishery;	Hooper et al. (2015);
		Sensitivity of rays, eels, cod, plaice to electromagnetic field of OWF	Andersson and Öhman (2010); Hooper and Austin (2015) Gill et al. (2005); Gill et al., 2012
Regulating services	Nutrient decomposition, gas regulation, etc.	Very limited studies and no relevant results	
		Abundance at OWF foundations is likely to increase the capacity of the system to remediate waste	Wilhelmsson and Malm (2008)
		Mainly studies on community structure and diversity of habitats, behavioural changes of species and changes in abundance of species associated with construction and operation	Brandt et al. (2011); Skeate et al. (2012);
Cultural services	Seascape and cultural services	Valuation of gain and loss changes for local residents in cultural services provided by seascape of the west coast of Schleswig-Holstein (Germany) in a context of OWF development. Offshore location of development limits the loss of amenity/aesthetic values,	Gee and Burckard (2010)

		but not the threat to nature and the symbolic value of the sea	
	Values of preferences	Change in preferences for OWF in relation to onshore/offshore location of the development. Offshore location of development limits the loss of cultural values measured as WTP	Ladenburgh (2009), Ladenburgh and Lutzeyer (2012); Voke et al. (2013)
Biodiversity	Scientific impacts to biodiversity	Contradictory results on animal impacts show both detrimental effects and limited direct impacts to avian population	Inger et al. (2009); Hattam et al. (2015)

Table 2: Different signals (defined using Habermassian categories in bracket) from the civil society for the three ecosystem services considered in the proposed examples addressing changes at local scales under the imperfect possibility to use only an economic approach to deal with the complexities of ecosystem services provision.

Type of Signal ➔ **Ecosystem Services ➔**	**Economic [money-steered/markets]**	**Social [lifeworld/deliberative]**	**Institutional [power-steered]**
Assimilative capacity	Reduced damages by pricing pollution, but in case of overtaking AC threshold.	Sense of danger for pollution amongst stakeholders.	Clear definition of property/user rights in negotiating how to reduce pollution.
Coastal defence	Valuation for assessing avoided damages and social-economic impacts of protection provided by natural dynamics	Sense of non-protection (insecurity felt by communities) provided by natural habitats	Use of environmental CBA of coastal defence, including externalities and benefits from ES
Energy from renewables	Mobilising capitals into the industry/valuation via EIA, financial discounted cash flow and CBA	Failure in science in providing the right information on MRE impacts on habitats and species	Precautionary principle applied in EU environmental policies

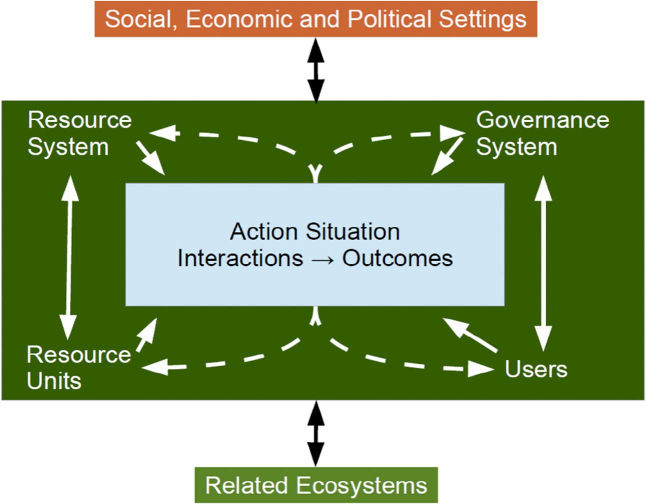

Social, Economic and Political Settings
Resource System
Governance System
Action Situation
Interactions → Outcomes
Resource Units
Users
Related Ecosystems

Figure 1

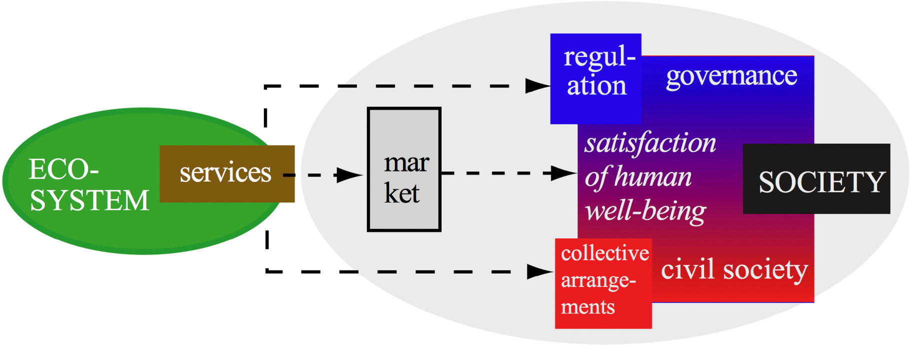

ECO-
SYSTEM
services
mar
ket
regul-
ation
governance
satisfaction
of human
well-being
SOCIETY
collective
arrange-
ments
civil society

Figure 2

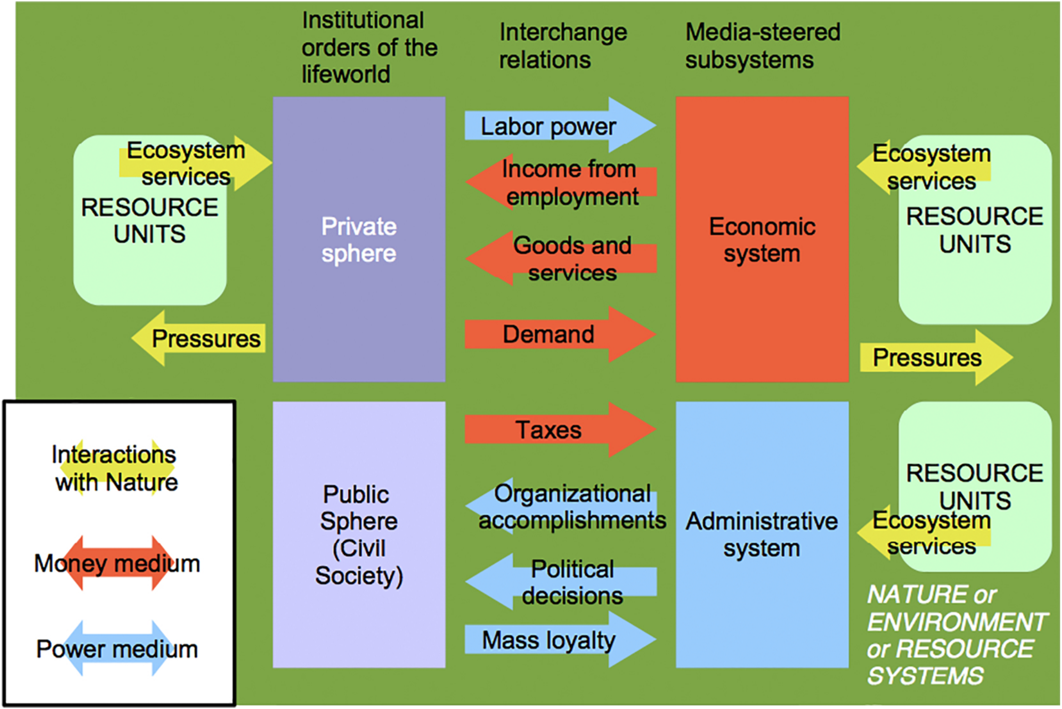
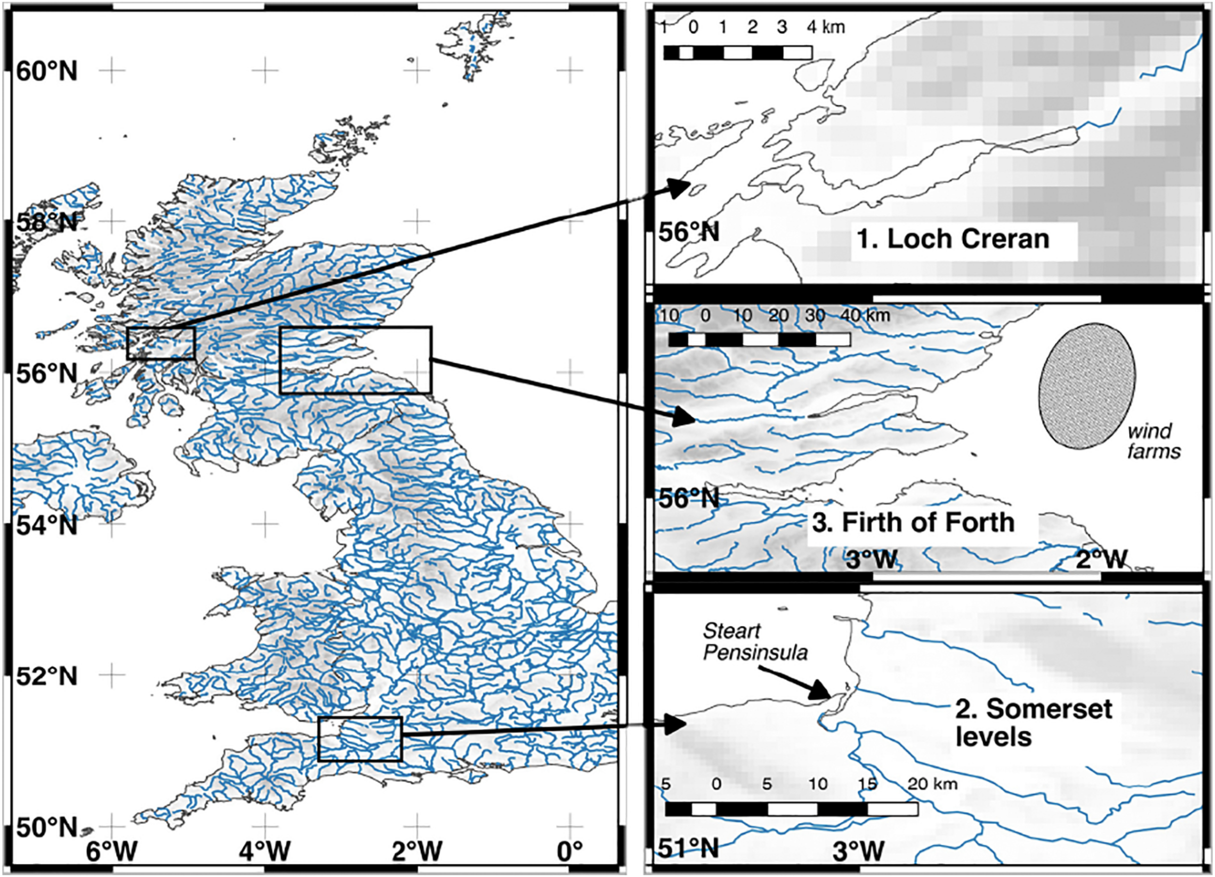
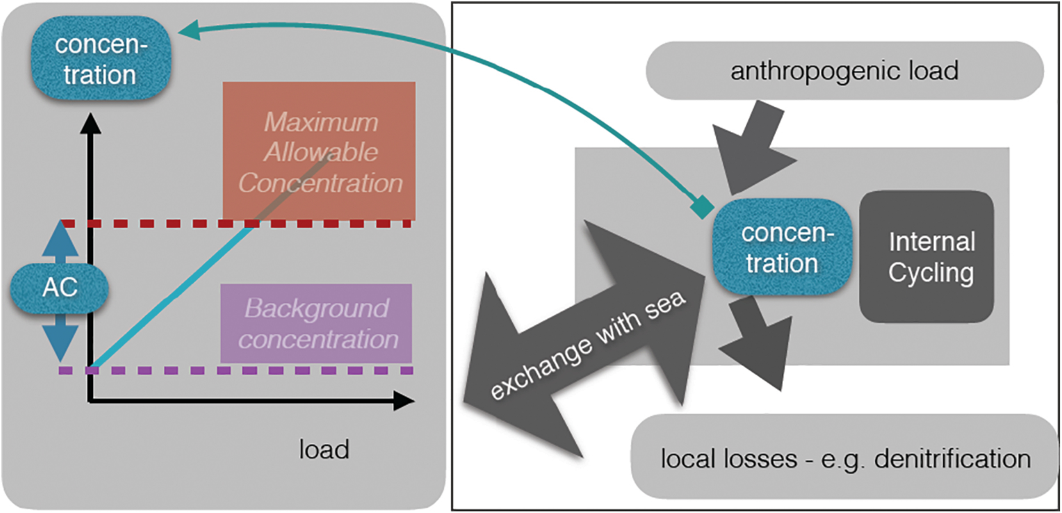

Institutional orders of the lifeworld
Interchange relations
Media-steered subsystems
Ecosystem services
RESOURCE UNITS
Private sphere
Labor power
Income from employment
Goods and services
Demand
Economic system
Ecosystem services
RESOURCE UNITS
Pressures
Pressures
Interactions with Nature
Money medium
Power medium
Public Sphere (Civil Society)
Taxes
Organizational accomplishments
Political decisions
Mass loyalty
Administrative system
RESOURCE UNITS
Ecosystem services
NATURE or ENVIRONMENT or RESOURCE SYSTEMS

Figure 3



60°N
58°N
56°N
54°N
52°N
50°N
6°W
4°W
2°W
0°
1 0 1 2 3 4 km
56°N
1. Loch Creran
10 0 10 20 30 40 km
wind farms
56°N
3. Firth of Forth
3°W
2°W
Steart Pensinsula
2. Somerset levels
5 0 5 10 15 20 km
51°N
3°W

Figure 4


concen-
tration
Maximum
Allowable
Concentration
AC
Background
concentration
load
anthropogenic load
concen-
tration
Internal
Cycling
exchange with sea
local losses - e.g. denitrification

(c)

Figure 5

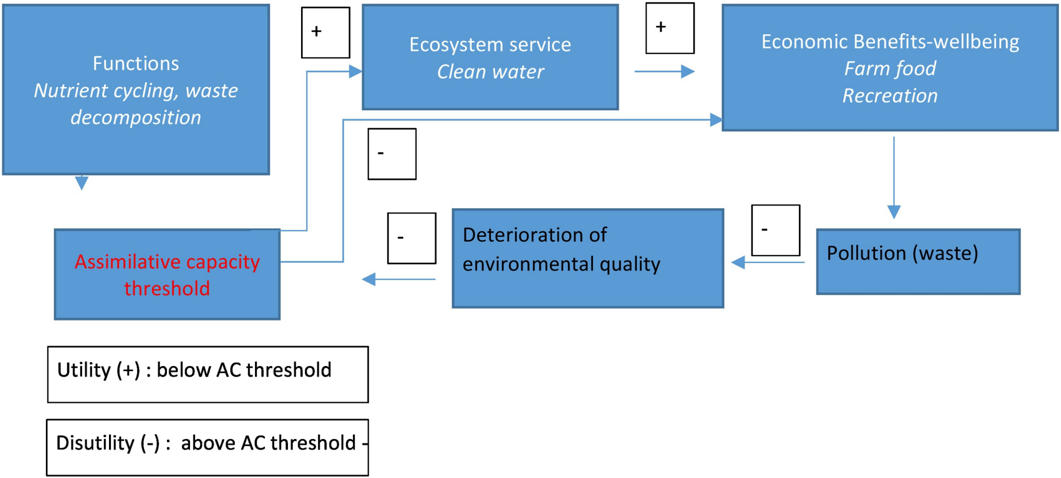

Functions
Nutrient cycling, waste decomposition
+
Ecosystem service
Clean water
+
Economic Benefits-wellbeing
Farm food
Recreation
-
Assimilative capacity threshold
-
Deterioration of environmental quality
-
Pollution (waste)
Utility (+) : below AC threshold
Disutility (-) : above AC threshold -

Figure 6

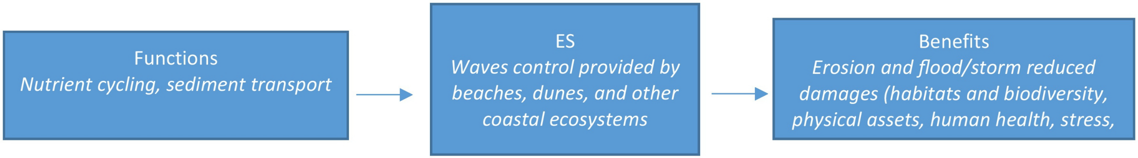

Functions
Nutrient cycling, sediment transport
ES
Waves control provided by beaches, dunes, and other coastal ecosystems
Benefits
Erosion and flood/storm reduced damages (habitats and biodiversity, physical assets, human health, stress,

Figure 7

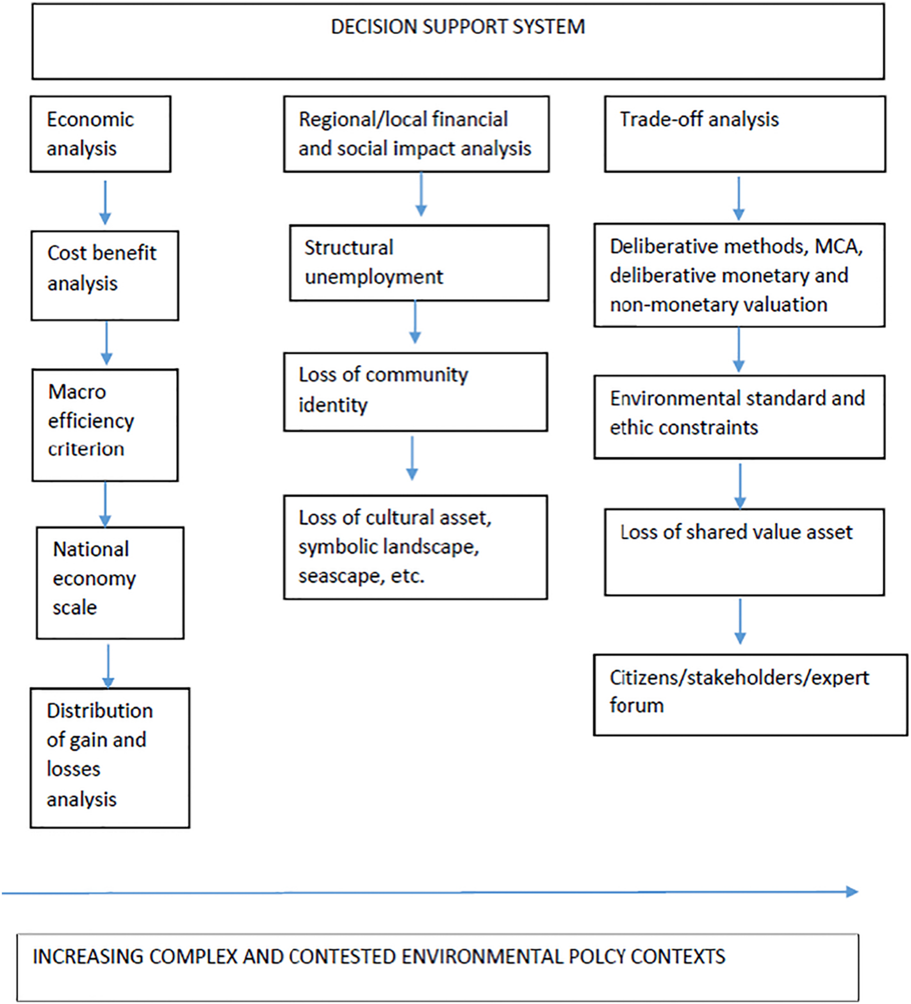

DECISION SUPPORT SYSTEM
Economic analysis
Cost benefit analysis
Macro efficiency criterion
National economy scale
Distribution of gain and losses analysis
Regional/local financial and social impact analysis
Structural unemployment
Loss of community identity
Loss of cultural asset, symbolic landscape, seascape, etc.
Trade-off analysis
Deliberative methods, MCA, deliberative monetary and non-monetary valuation
Environmental standard and ethic constraints
Loss of shared value asset
Citizens/stakeholders/expert forum
INCREASING COMPLEX AND CONTESTED ENVIRONMENTAL POLCY CONTEXTS

Figure 8